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Glass Fibre Composites Recycling Using the Fluidised Bed: A Comparative Study into the Carbon Footprint in the UK

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Abstract: The UK has no established process for recycling waste glass fibre-reinforced thermosets that are widely used within wind blade structures. Consequently, these materials are typically disposed of in landfills or undergo energy recovery in waste facilities. This study investigates the carbon footprint of the fluidised bed process for recycling glass fibre composite waste, considering the present and future scenarios of composite waste management in the UK. The impact was compared to conventional disposal routes and other prominent recycling technologies, such as cement kiln co-processing and mechanical recycling, by developing energy and material flow models for each waste treatment strategy. Variables, such as the type of waste, the quantity of recycling facilities in the UK, and waste haulage distance, were examined to inform the lowest impact deployment of recycling technologies. Cement kiln co-processing, mechanical, and fluidised bed recycling technologies reduced the global warming potential of processing wind blade waste compared with conventional disposal routes, with impacts of −0.25, −1.25, and −0.57 kg CO_2e/kg GRP waste, respectively. Mechanical recycling had the lowest global warming potential resulting from low greenhouse gas emissions associated with the process itself and potentially high offsets by replacing glass fibre in the production of moulding compound. Composite wind turbine blade waste was found to be a particularly promising feedstock for the fluidised bed process due to relatively low resin content diminishing direct greenhouse gas emissions during thermal decomposition, as well as high material recovery offsets due to the high glass fibre content of this waste stream.

Keywords: composites recycling; glass fibre recycling; fluidised bed; mechanical recycling; cement kiln co-processing; environmental assessment

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1. Introduction

One of the foremost challenges confronting the composites industry today is the environmentally responsible disposal of composite products at the end of their lifecycle. The annual global production of fibre-reinforced polymer composites has surpassed 12 million tons (Mton, 10^9 kg in SI units) [1] and is expected to grow to 15 Mton by 2025 [2]. The European composites market is approaching 3 Mton per year [1], with annual glass fibre-reinforced plastics (GRP) production volume in Europe alone in excess of 1 Mton [3]. Thermoset-based polymer composites constitute roughly 45% of the market, while glass fibre-reinforced composites make up over 90% of the entire fibre-reinforced composite production [1]. The substantial demand for GRP leads to the generation of a significant volume of waste from composite production and end-of-life (EoL) products. In particular, the disposal of EoL wind turbine blades (EoL–WTB), primarily constructed from GRP, is anticipated to see a substantial increase in global waste over the forthcoming decades. Projections suggest that by 2030, this waste is expected to approach 0.5 Mton per year and grow to 1 Mton per year by 2040 [4]. Furthermore, it is estimated that the current annual GRP EoL waste in the UK alone has reached 50–60 thousand tons (kton, 10^6 kg in SI units), with an additional 15 kton per year in GRP production waste [5]. A recycling process capable of extracting glass fibres from GRP waste, which can replace new fibres in the production of GRP, could have the benefits of reducing both the number of composite materials landfilled as well as saving resources required to manufacture virgin glass fibres (vGF).

Thermoset-based GRPs cannot easily be reused/recycled due to their molecular crosslinking. Extensive research has been recently devoted to the development of composite recycling techniques, which have led to various recycling strategies, each at different Technology Readiness Levels (TRL). In recent years these technologies have been widely reviewed in terms of the methods under development [6,7], TRL [8], recovered material properties [9,10], ongoing challenges [11], optimisation pathways [12], and environmental impacts [13,14].

The prominent technologies can be broadly categorised into three types of recycling processes: thermal, mechanical, and chemical. Each process aims to extract material and/or energy from the GRP waste that can be reused to replace the production of raw materials/offset energy demand. The processes typically aim to extract the glass fibre fraction within the waste GRP to then reuse as a reinforcement medium in new materials, offsetting the relatively high energy demand in the production of vGF. Among them are combustion-based thermal recycling methods such as cement kiln co-processing and the fluidised bed recycling (FBR) process, each of which can utilise energy from the polymer fraction of GRP waste as well as reuse inorganic fractions to offset raw material production [15,16]. The thermal recycling of glass fibre-reinforced thermosets through the FBR process has shown numerous benefits, including scalability, continuous operation, tolerance to contaminants, the ability to process diverse polymers, and the production of fibrous materials without char [17–19].

Mechanical recycling involves downsizing/grinding the waste GRP and then using grading technology(s) to classify the waste into factions with different sizes/compositions. Fibre-rich fractions can be used in place of vGF in GRP production, and finer powder fractions can be used as fillers [20]. Chemical recycling processes use heated solvent/solvent mixtures to break down the thermoset into smaller weight molecules, allowing for the recovery of glass fibres with the potential for reclaiming materials from the depolymerised resin in the form of monomers or petrochemical feedstocks [5,21].

Given the global push toward Net Zero emissions targets, it is critical that low-carbon technologies are pursued for recycling GRP waste streams. In recent years, several authors have estimated the carbon footprint of various EoL strategies for GRP wind blade structures. Nagel et al. quantified the environmental impacts of disposing of blade waste in Ireland using Life Cycle Assessment (LCA) [22]. The study compared co-processing blade waste in a cement kiln with a landfill, concluding that cement kiln co-processing could reduce the contribution to human health, ecosystem toxicity, climate change, and resource impacts [22].

Cong et al. used LCA to measure the carbon reduction potential of four disposal scenarios of EoL GRP wind blades in China [23]. The methods analysed included incineration in a waste-to-energy plant, cement kiln co-processing, and two similar methods to utilise GRP recyclates for replacing cement and aggregates in construction and infrastructure applications. Cong et al. determined that utilising blade waste in these applications has the potential to reduce their respective carbon footprints, concluding that recycling strategy can reduce the lifecycle carbon emission intensity of a given wind farm from 12.51 g CO_2e/kWh to 9.22 g CO_2e/kWh.

Yang et al. assessed the environmental and financial costs associated with various waste treatment methods for wind turbine blade waste in China [13]. Both conventional approaches, such as landfill and incineration, as well as contemporary handling and recycling technologies like fluidised bed, pyrolysis, and mechanical and chemical recycling, were considered. The findings of Yang et al. indicated that recycling wind turbine blades leads to a reduction in greenhouse gas (GHG) emissions compared to traditional landfill and incineration methods, except in the case of pyrolysis. The overall environmental impact is influenced by the quantity of waste and the carbon intensity of electricity grid mixes. As such, the generalisability of these results to other regions, such as Europe, is limited by regional disparity in electrical grid mix impact. Therefore, it is critical to conduct assessments for each region to determine localised low-carbon strategies.

Sproul et al. conducted a comparative LCA to assess GHG emissions and material yields from a range of wind turbine blade recycling approaches (cement co-processing, mechanical, pyrolysis, microwave pyrolysis, and solvolysis recycling) in the U.S [14]. Sproul et al. determined that both mechanical recycling and microwave pyrolysis result in the lowest net GHG emissions. Nevertheless, the reliability of the value of mechanically recycled materials is subject to significant uncertainty. This is due to the mixed feedstock generated by mechanical recycling, which may exhibit inferior performance compared to virgin materials.

While several authors have investigated the carbon footprint of GRP wind blade recycling in isolation, blade waste constitutes a minority of the total volume of GRP waste both in the UK and globally. It is therefore critical to also understand the environmental impact of recycling GRP from other, larger volume waste streams. Moreover, to maintain economic viability (at least in the short term when blade waste volumes are low), it is likely that wind blades will be co-processed with other GRP waste streams [24]. As such, this study investigates the carbon footprint of the thermo-oxidative FBR process for GRP recycling within the prevailing and anticipated landscape of mixed GRP waste in the UK. This is compared to the environmental impact of conventional waste disposal routes (landfill and energy from waste) and higher TRL GRP recycling processes (cement kiln co-processing and mechanical). This work is critical to deploying sustainable GRP waste treatment methodologies and contributes to the United Nation's Sustainable Development Goals by promoting more resource-efficient practices and giving recommendations to enable the mitigation of GHG emissions and climate change.

2. Methods

2.1. Goal and Scope

The aim of this assessment is to evaluate the global warming potential (GWP) of the thermo-oxidative fluidised bed process for the recycling of GRP waste and to assess the influence of various GRP waste streams, the scale of recycling plants and their distribution across the UK was assessed. Additionally, the GWP of other EoL strategies are evaluated and compared in order to identify low-carbon solutions for GRP waste streams.

2.1.1. Functional Unit

This study uses the following functional unit: 1 kg of GRP waste recycling with any of the material compositions described in the study (e.g., as described in Figure 1). The reference flow is 1 kg of GRP waste.

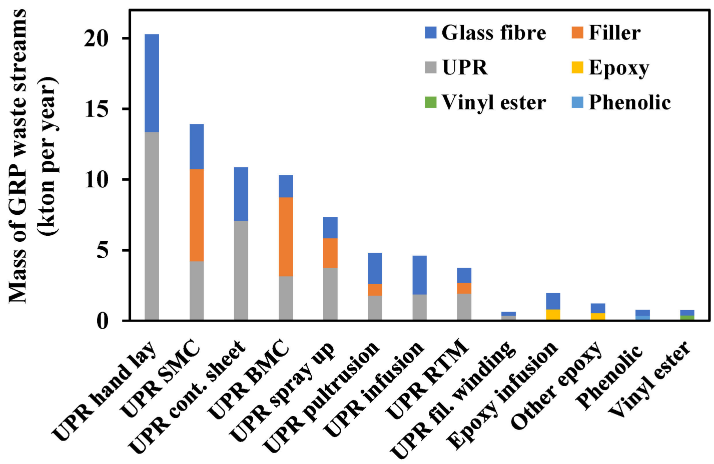

Figure 1. Mass of GRP varieties and their component materials in the waste stream in the UK, generated from data presented in [25] and reproduced in [24].

2.1.2. GRP Waste Stream Scenarios

The current and prospective carbon footprints of GRP waste processing in the UK were evaluated by examining the waste stream in terms of both the quantity and composition of GRP. Three distinct waste streams were considered:

(1) Current UK mixed GRP waste;
(2) GRP from projected UK end-of-life WTB waste (EoL–WTB);
(3) GRP from combined UK end-of-life WTB waste and UK GRP manufacturing waste (EoL–WTB + Manf.).

Current UK Mixed GRP Waste

Composites UK reported estimates for the quantity and composition of various types of GRP waste streams in the UK [22]. Waste volumes were estimated for GRP for predominant composite formats and resins. The estimated data are anecdotal, based on interviews with experienced industry professionals, but serve as a good basis for estimating available waste volumes. The methodology used is described in more detail in [25]. Figure 1 shows the estimated mass of GRP types and constituent materials in the UK waste stream (UPR—unsaturated polyester resin, SMC—sheet moulding compound, BMC—bulk moulding compound, RTM—resin transfer moulding) [25]. For the purposes of this research, it is assumed that the "filler" material exclusively consists of calcium carbonate ($CaCO_3$), as specific quantities for individual filler types are not specified in [25]. $CaCO_3$ exhibits mass consistency during processing and can be directly substituted for new $CaCO_3$ when reused.

"Mixed GRP waste" is characterised as GRP waste that encompasses all types of GRP in the proportions specified in [25], as depicted in Table 1.

Table 1. Composition of total UK GRP waste stream defined as "Mixed GRP waste", generated from data presented in [25] and reproduced, in part, from [24].

Material	Mass in UK GRP Waste (kton per Year)
Glass fibre	25.8
Filler ($CaCO_3$)	15.8
UPR	37.6
Epoxy	1.36
Vinyl ester	0.38
Phenolic	0.38
Resin total	39.7
GRP total	81.3

EoL–WTB

Establishing a recycling facility dedicated to a single GRP waste stream offers the advantage of maintaining a consistent feedstock, resulting in predictability in operations and the materials it produces. Waste WTB may be managed with relative ease, bypassing the complexities associated with segregating, classifying, and managing mixed GRP waste streams. Considering the consistent characteristics of WTB waste, establishing a recycling facility utilising EoL–WTB exclusively as its primary input could potentially offer solutions to numerous practical challenges associated with mixed GRP recycling. Figure 2 shows the projected GRP mass in UK EoL–WTB by country used in this study, provided by Zero Waste Scotland, which is reported for Scotland specifically in [26]. As illustrated in Figure 2, there is a notable projection of the GRP mass originating from EoL–WTB, signifying its rapid emergence as a substantial source of waste GRP in the UK. According to [27], EoL–WTB is assumed to consist of 60% by weight glass fibre, 32% by weight epoxy, and the remaining 8% by weight comprises materials such as paint, adhesive, foam, and balsa.

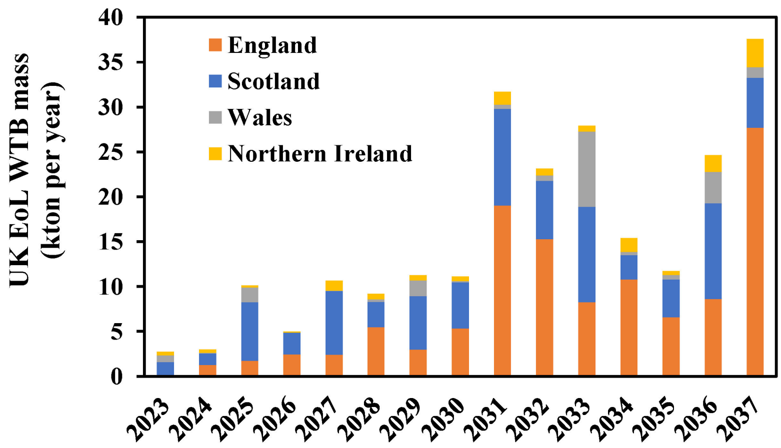

Figure 2. EoL–WTB waste mass projections by UK country, reproduced from [24].

EoL–WTB + Manf.

Annually, the UK generates approximately 15 kton of waste GRP, which is estimated to represent around 20% of the mass of waste GRP [5]. Compared to other GRP waste types, manufacturing waste and EoL–WTB are expected to reduce sorting demands when not previously integrated into mixed material supply chains, rendering them an appealing option for recycling facilities. In this study, "EoL–WTB + Manf." represents the combination of EoL–WTB and manufacturing waste. It is presumed that manufacturing waste shares the same composition as the mixed GRP waste in the UK, with a constant annual mass of 15 kton.

2.1.3. EoL Treatment Scenarios

In this study, five GRP waste treatment scenarios are investigated:

(1) Landfill;
(2) Energy from waste (EfW);
(3) Mechanical recycling;
(4) Cement kiln co-processing;
(5) Fluidised bed recycling.

Energy models for the processes were developed to assess the energy demand and GWP associated with each waste treatment option. Five stages of waste processing, shown in Figure 3, were considered, which do not necessarily apply to all solutions analysed: (1) GRP pre-treatment, (2) GRP transport, (3) GRP treatment, (4) residue transport, and (5) residue landfill. The waste treatment methods are described in detail in Section 2.4, outlining the assumptions in generating inventory data for the analysis.

It should be noted that the set of recycling strategies assessed in this work is not exhaustive of all proposed technologies, outlined for example in [28]. FBR was selected for primary analysis based on the author's access to operational data needed to conduct a robust analysis, which has been compared to other high TRL recycling technologies where input data are available in the literature (mechanical recycling and co-processing). Other chemical- or thermal-based recycling technologies come in a variety of configurations, and it is not within the scope of the study to analyse all these technologies.

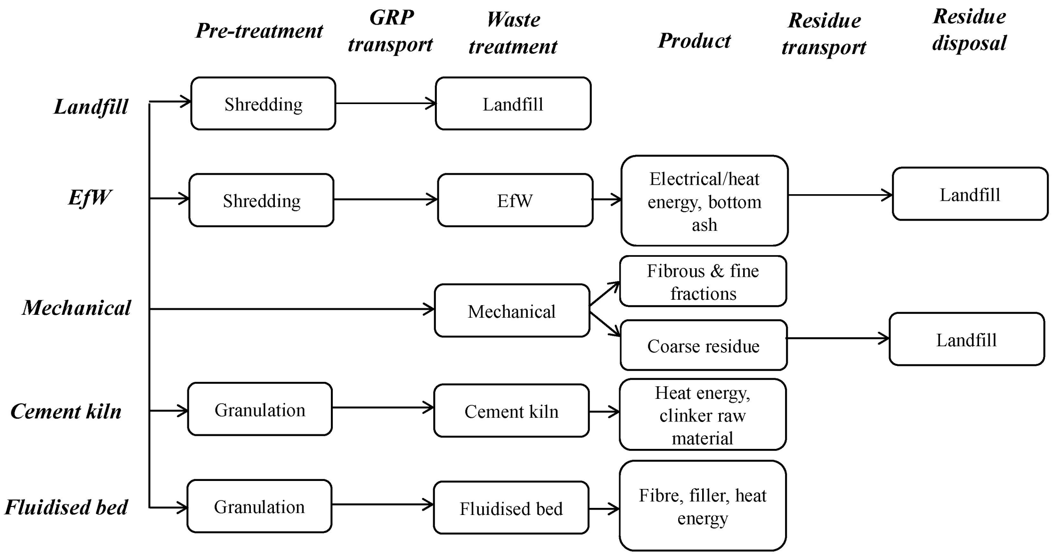

Figure 3. Overview of the stages of processing for each of the six GRP waste treatments considered.

2.1.4. System Boundary

All impacts associated with GRP manufacture, use, and decommission are outside the system boundary of the study, as shown in Figure 4. As such, no burden is attributed to the GRP feedstock entering the system boundary and used in the production of secondary materials. Figure 4 shows the system boundary and the processes that are attributed to the GRP end-of-life treatment/recycling phase.

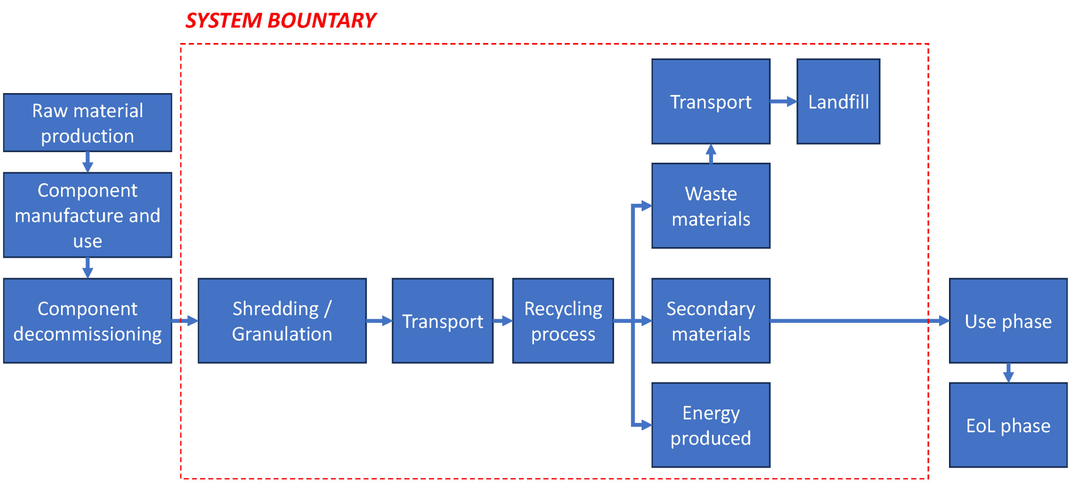

Figure 4. System boundary of assessment.

2.2. *Environmental Impact Indicators*

This study concentrates on appraising the carbon footprint associated with different EoL GRP treatment strategies; therefore, the global warming potential (GWP 100 years) was evaluated and measured in kg of CO_2 equivalent. For a complete environmental assessment, it is pertinent to also consider other environmental impact indicators, given that some strategies may be more or less favourable depending on the indicator of concern. Given the pressing concern posed by GHG emissions and their potential impact on environmental and ecological stability, the scope of this study was limited to evaluating GWP; however, prior to the industrialisation of recycling technology(s), other impact indicators should be evaluated.

2.3. Allocation

In scenarios where the GRP is recycled at the end of its life, there are always one or more secondary material products produced during the recycling process. The following are allocated to the GRP end-of-life phase:

- The burden of the GRP recycling/secondary material production;
- The burden of disposing of waste by-products;
- The avoided burning of all secondary material products (recycled GRP materials);
- The avoided burning of energy generation, where energy is an output of the EoL treatment (e.g., energy from waste).

By using the avoided burden approach, the following are observed:

- The burden for virgin materials used is allocated to the system;
- The burden of secondary material production is included within the system boundary and is allocated to the system;
- Materials recycled at end-of-life offset the demand for a quantity of virgin material;
- All GRP waste is either recycled or disposed of in the system boundary, and avoided burden credit is given to secondary materials produced.

The net GWP of each waste GRP treatment was calculated in Equation (1) by summing the GWP of each stage of the process and deducting the GWP offsets provided by the treatment (power generation/heat recovery and virgin materials).

$$\text{Net GWP} = \sum \text{GWP}_{\text{output}} - \sum \text{GWP}_{\text{offset}} \quad (1)$$

2.4. Assumptions in Inventory Calculations

2.4.1. GRP Pre-Treatment

For the purposes of this investigation, the only pre-treatment considered is material downsizing. For landfill and energy from waste scenarios, GRP is required to be roughly downsized using a shredder with an energy demand of 0.09 MJ/kg GRP [29]. FBR and cement co-processing required additional downsizing of GRP to 5–25 mm and <40 mm [22], respectively, which is achieved using a granulator with energy demand calculated using data and methodology outlined in [30].

2.4.2. GRP Transport

In all scenarios, it is assumed that waste is transported using a 32-ton diesel heavy goods lorry with a 20-ton load capacity, meeting Euro VI standards [31] with fuel consumption of 31 litres per 100 km and emissions of 0.830 kg of CO_2 equivalent per km at a cruising speed of 70 km per hour [32].

The study considers transportation distances for waste GRP before recycling, which depend on the amount and distribution of recycling facilities across the UK. Transportation distances are calculated based on an even distribution of recycling plants across the mainland UK. The assumptions used to estimate transport distances to recycling facilities have been previously detailed by the authors in [24]. Transportation distances to landfill and energy from waste plants were assumed to be constant throughout the study at 100 km and 200 km, respectively, given their ubiquity across mainland UK.

2.4.3. GRP Waste Treatments

Landfill

Disposing of GRP waste in landfills is assumed to occur in a sanitary landfilling site, which is built for the final disposal of solid waste. Due to the inert nature of GRP waste, it is assumed that no significant direct emissions are associated with landfilling. The energy demand for general landfilling operations is assumed to be 0.167 MJ/kg GRP with energy split between the UK electricity supply and fossil fuels [33].

EfW

Incinerating GRP waste extracts chemical energy from the resin fraction, converting it into usable electrical power and, in some cases, heat. Co-processing GRP waste with solid municipal waste in EfW facilities is common, but the process does not reclaim non-combustible filler and fibre fractions, making it not a true recycling method. Energy production depends on GRP waste energy content and incineration plant efficiency. Figure 5 depicts EfW processes and energy input/output sources in the model.

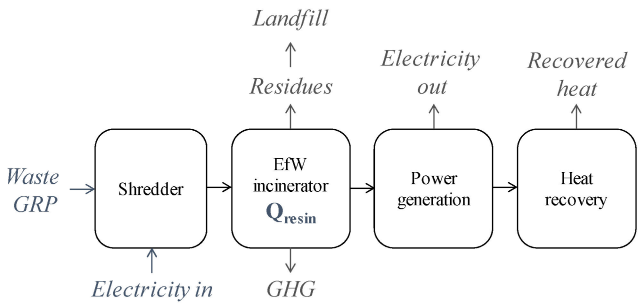

Figure 5. Schematic showing the stages of EfW, including energy inputs through resin combustion (Qresin) and electrical energy inputs and outputs.

The calorific value of GRP depends on resin systems and the resin fraction. EfW facilities vary in efficiency (typical electrical efficiencies: 15–24% [34]). Plants in combined heat and power mode can reach over 70% efficiency by using waste heat [34]. For UK waste GRP solutions, the model uses the average efficiency across all UK energy-from-waste plants. Only 10 out of 48 UK energy-from-waste plants recover heat, with most generating only electrical power [35]. The UK averaged 531 kWh per ton of electrical power and 110 kWh per ton of heat in 2019, resulting in overall efficiencies of 20.7% and 4.3%, respectively [35]. Assuming 80% efficiency, generated electricity displaces the UK mains supply, and heat offsets natural gas combustion. While bottom ash from energy-from-waste plants can be reused in construction, the rate and specifics of this practice are unclear [36]. Thus, it is assumed that residual filler and fibre fractions of waste GRP are landfilled. Assumptions in estimating elemental content, calorific value, and emissions produced as a result of oxidation of the various resin systems are described in [24], and data are tabulated in Table 2.

Table 2. Elemental content, calorific value, and emissions produced as a result of oxidation of the various resin systems as described in [24].

Resin	Fraction of Resin (%wt.)				Calorific Value (MJ/kg Resin)	GWP_{resin} (kg CO_2e/kg Resin)
	Carbon	Hydrogen	Oxygen	Nitrogen		
UPR	71.0	6.1	22.9	0	30.2	3.30
Epoxy	73.6	7.8	16.4	2.2	33.9	2.60
Vinyl ester	80.6	7.4	12.0	0	36.6	2.95
Phenolic	66.9	6.7	21.2	5.2	29.2	3.18

Mechanical Recycling

The mechanical recycling simulation for GRP encompassed a downsizing and classification process detailed in [20] and depicted in Figure 6.

Initially, GRP is ground using a granulator and subsequently classified into four fractions employing a "zig-zag" classifier. The fractions have varying compositions and sizes, including two fibre-rich fractions, a power fraction, and a coarse resin-rich fraction, as described in Table 3.

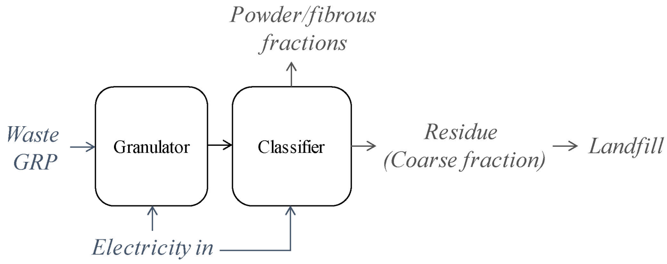

Figure 6. Schematic showing the stages of mechanical recycling, including electrical energy inputs and material output.

Table 3. GRP fractions after classification stage of mechanical recycling and the relative change in composition for each.

Fraction	Proportion of GRP (%)	Rise in Fibre/Resin Ratio (%)	Rise in Filler/Resin Ratio (%)	Rise in Resin Fraction (%)	Use
Powder	30	4.3	33.4	−7.3	Filler
Fibrous A	21	58.4	14.9	−17.6	Fibre (BMC/SMC)
Fibrous B	21	84.2	16.5	−23.0	
Coarse	28	−59.8	−33.8	34.3	N/A (Landfilled)

Energy requirements for both downsizing and classification stages in mechanical recycling were modelled, with the energy for downsizing, achieved through a granulator, calculated following the method in [30]. The energy demand for "zig-zag" classification, influenced by equipment size, flow rates, and particulate loading, was modelled and experimentally measured in [37]. A nominal energy demand of 5 kJ/kg GRP for classification was determined based on an average of measured data in [37], necessitating three repetitions to obtain the four GRP fractions. All energy utilized in mechanical recycling is assumed to be electrical and sourced from the UK's mains supply.

Palmer demonstrated that the two fibre-rich fractions ("Fibrous A" and "Fibrous B" in Table 3) could substitute 10% of vGF mass in bulk/sheet moulding compound (BMC/SMC) production without compromising GRP strength [38]. It is assumed that the recovered fibre-rich fractions can counterbalance the production of an equivalent mass of vGF, commonly used in BMC/SMC product manufacturing. Given the observed lack of impact on GRP performance in such applications, it is reasonable to presume parity in the replacement rate of vGF using mechanically recovered fibre-rich fractions. Additionally, the recovered powder fraction is presumed to counteract the production of an equivalent mass of $CaCO_3$ filler products. However, no identified use exists for the coarse, resin-rich fraction [38], which is assumed to be landfilled in this study.

Cement Kiln Co-Processing

GRP waste is already commercially used as both fuel and raw materials in the production of clinker within cement kilns [39]. The production process combines raw material fractions with an energy source (fossil fuel) and is heated to around 1450 °C to produce cement clinker, which is composed of calcium oxide, silica, alumina, and iron oxide. Combustion of the organic fraction of the GRP waste within cement kilns can be used to heat the process and offset demand for other fuels typically used in the process, such as coal. In this study, it is assumed that GRP waste can replace coal at an energy equivalence basis following Equation (2).

$$Coal\ replacement\ rate = \frac{m_{coal}}{m_{GRP}} = \frac{CV_{GRP}}{CV_{coal}} \tag{2}$$

E-glass with silica, calcium oxide, and alumina content of 54, 18, and 15%wt., respectively, was used in modelling the co-processing route. It was assumed that all minerals would directly replace raw materials on a mass equivalency basis, offsetting the energy and emissions associated with the upstream production of these materials. Figure 7 shows a schematic of the cement kiln co-processing energy model, including energy input through resin combustion, electrical energy input, and recovered materials.

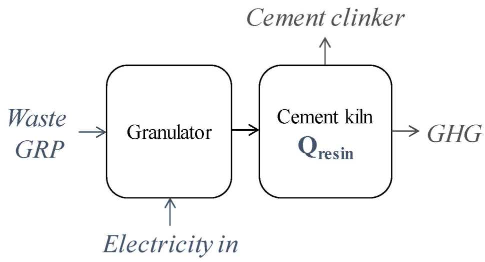

Figure 7. Schematic showing the stages of cement kiln co-processing, including energy input through resin combustion (Qresin), electrical energy input, and recovered materials (clinker).

Fluidised Bed Recycling

A lab-scale fluidised bed recycling system situated at the University of Strathclyde demonstrated the successful recycling of GRP from WTB scrap [15]. The recycled glass fibres have been used in the production of a 3 kW domestic wind blade, as depicted in Figure 8. A detailed description of this laboratory-scale fluidised bed can be found in the author's previous publication [40]. The University of Strathclyde is presently in the phase of scaling up this technology as an integral component of a UK groundbreaking wind turbine blade recycling pilot plant [41].

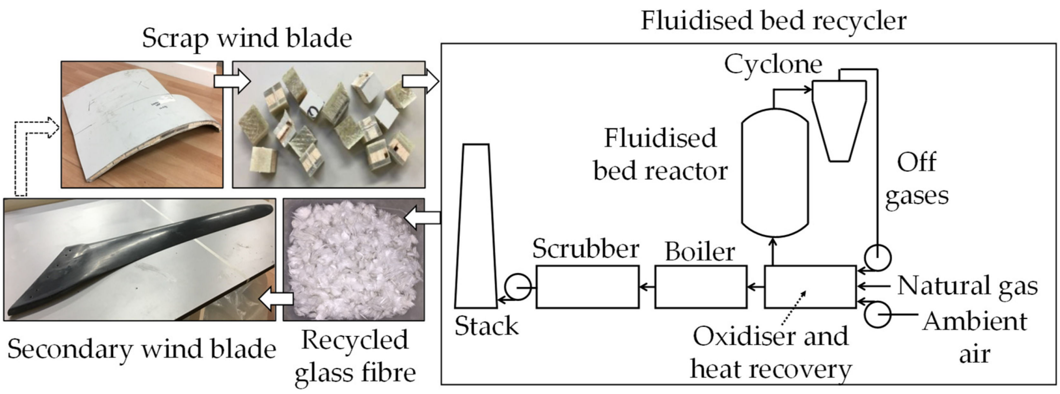

Figure 8. WTB recycling using the fluidised bed at the University of Strathclyde described in [40] and reproduced from [24].

The experimentally determined operating parameters for the fluidised bed were incorporated into energy models designed for a commercial-scale FBR process. The schematic of the proposed FBR plant is illustrated in Figure 8. The reactor is maintained at a temperature of 550 °C to expedite the decomposition of the polymer matrix, freeing the clean filler and fibrous components, which are entrained out of the reactor by the fluidised gas and subsequently separated and collected. Combustion gases undergo full oxidation to eliminate volatile gas components and subsequently pass through a sequence of high and low-temperature heat exchangers, facilitating the recovery of heat for reuse within the process. A boiler is positioned before the stack to generate a process stream for internal use or local sale. Forced and induced draft fans are utilized to maintain a sufficient flow within the system and to overcome pressure losses across the different components.

To generate the required input into the environmental assessment, an energy model was devised, which was informed by the operation of the University of Strathclyde's FBR process. In the model, heat is introduced in the reactor by oxidising the organics in the GRP feedstock and through the oxidation of natural gas in the oxidizer. The volume of natural gas needed to sustain the reactor's temperature (550 °C) was determined by balancing heat inputs and the losses in the stack and conductively through the system's insulation. The electricity consumed by the fans was predicted by evaluating the required fluidisation gas flow rate and pressure rises throughout the system.

The variable operating parameters are used to determine the required reactor cross-sectional area of the fluidised bed reactor, as described in Equation (3).

$$Reactor\ area\ [m^2] = \frac{Installed\ Capacity\ \left[\frac{kg\ GRP}{yr}\right] \times GF\ weight\ fraction\ \left[\frac{kg\ GF}{kg\ GRP}\right]}{Operating\ time\ \left[\frac{hr}{yr}\right] \times Reactor\ loading\ rate\ \left[\frac{kg\ GF}{hrm^2}\right]} \quad (3)$$

The superficial air velocity passing through the fluidised bed (fluidisation velocity) and flowing through the pipes is set to 1 and 20 m/s, respectively. From this, the gas flow rate through the system can be defined, and the other plant components are scaled accordingly.

Installed capacity—the annual GRP throughput capacity of the fluidised bed plant (kton of GRP per year).

Reactor loading rate—the glass fibre mass feed rate into the fluidised bed as a function of reactor cross-sectional area (kg GF/(h·m^2)), which was fixed at 10 kg GF/(h·m^2) throughout this study.

It is assumed that recovered glass fibre and filler materials can offset the production of an equal mass of new glass fibre and filler products. It is well established that glass fibre experiences performance loss during thermal recycling, limiting applications for direct replacement of new materials [15]. Kennerley et al. demonstrated that direct replacement of vGF with an equal mass of rGF in the production of the bulk moulding compound caused no adverse effect on the subsequent mechanical performance of the composite material produced [42]. For such applications, where no impact on GRP performance is observed, it is reasonable to assume parity in the replacement rate of vGF using rGF. A more exhaustive description of the assumptions made in the fluidised bed energy model is given in the authors' previous publication [24].

Table 4 shows a summary of the energy and GWP inputs used in modelling the various waste treatment processes.

Table 4. Summary of energy and GWP inputs used to model the various processes and in the environmental assessment, * incl. transmission and distribution losses.

	Embodied Energy	GWP
Virgin glass fibre [43,44]	22.5 MJ/kg	2.91 kg CO_2e/kg
$CaCO_3$ (GRP filler) [45]	4.3 MJ/kg	0.27 kg CO_2e/kg
UK grid electricity [46]	-	0.21 kg CO_2e/kWh *
Natural gas (Combusted) [46,47]	39.5 MJ/m^3	0.19 kg CO_2e/kWh
Diesel (Combusted) [48]	45.3 MJ/kg	0.25 kg CO_2e/kWh
Coal [49]	0.23 MJ/kg	0.19 kg CO_2e/kg
Silica [50]	0.10 MJ/kg	0.053 kg CO_2e/kg
$CaCO_3$ (Cement kiln raw material) [45]	0.32 MJ/kg	0.023 kg CO_2e/kg
Alumina [51]	14.5 kWh/kg	1.0 kg CO_2e/kg

2.5. Impact Assessment

To conduct the environmental assessment, energy and mass flow models were developed in MS Excel. Energy models for each treatment method were produced using assumptions outlined in Section 2.4 above, which enabled the electrical and heat energy consumption to be estimated relative to the reference flow (1 kg of GRP processed). Mass flow models were also produced based on assumptions outlined in Section 2.4, which enabled the mass flow of products, waste, and direct atmospheric emissions (e.g., from polymer combustion) to be quantified relative to the reference flow. A waste stream database was produced from data outlined in Figure 1, which was used as input in the energy and mass flow models, which enabled input data for the assessment to be determined for each treatment method and waste GRP feedstock type. The GWP was then calculated for each energy/material flow by scaling with inventory data presented in Table 4.

3. Results

3.1. Fluidised Bed Recycling of GRP Waste

3.1.1. Influence of GRP Waste Type

Figure 9 shows the magnitude and sources of GWP associated with recycling the range of GRP types within current UK GRP waste streams, using 20 kton of GRP per year per FBR plant. Approximately half of the waste types in Figure 9 present a negative net GWP, which indicates an overall reduction in equivalent CO_2 emissions through recycling. The major source of emissions is the FBR process itself, with direct GHG emission from resin decomposition accounting for 83–94% of the GWP of the process, depending on resin type and content in GRP waste. Additional heat recovered from higher resin content GRP in Figure 9 (such as "UPR hand lay" and "UPR cont. sheet") is insufficient to offset the greater emissions caused by resin oxidation within the fluidised bed. Material recovery is overwhelming the greatest GWP offset for the FBR across the GRP types analysed in Figure 9.

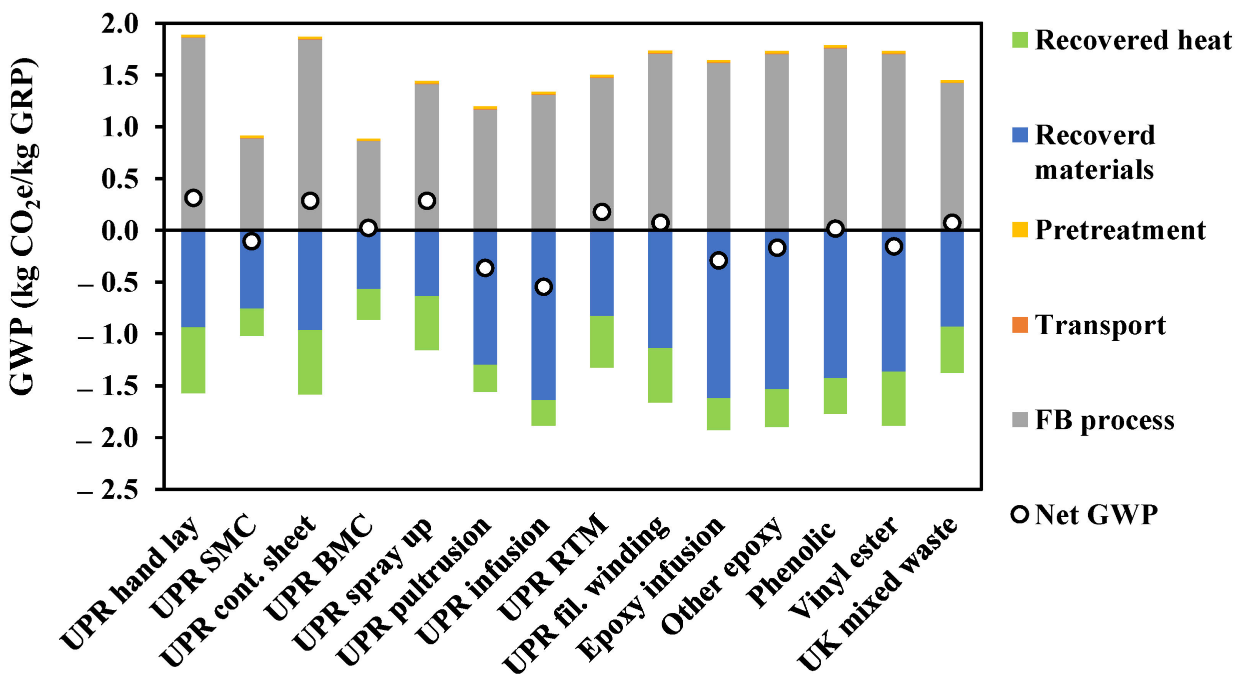

Figure 9. FBR GWP for range of GRP types and plant capacity of 20 kton of GRP per year.

Figure 10 compares the output GWP for UPR-based GRP using FBR as a function of resin weight fraction in the waste GRP. Output GWP (GWP prior to accounting for offsets through heat and material recovery) increases with the resin content of GRP in Figure 10, as would be expected given the additional resin mass required to be oxidised and subsequent GHG emissions this entails. Therefore, in terms of GWP, the fluidised bed process potentially has the greatest positive environmental impact when processing high fibre, low resin content GRP waste such as EoL–WTB.

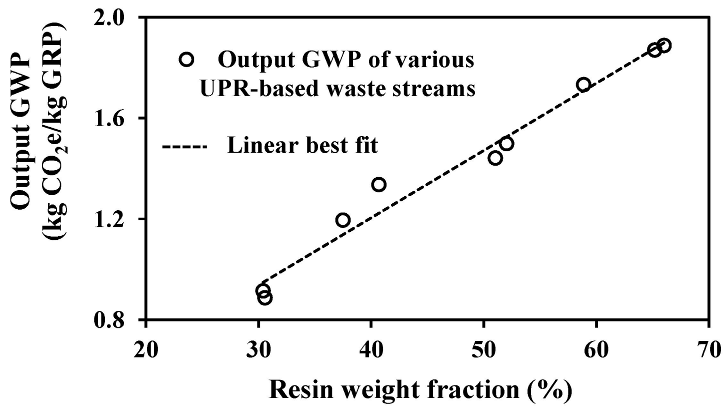

Figure 10. Output GWP when recycling URP-based waste streams using fluidised bed relative to the resin fraction in GRP.

3.1.2. Influence of Number of Plants

Figure 11 presents the resulting GWP when recycling "UK mixed GRP waste" in the FBR process for a range of plant capacities and number of plants.

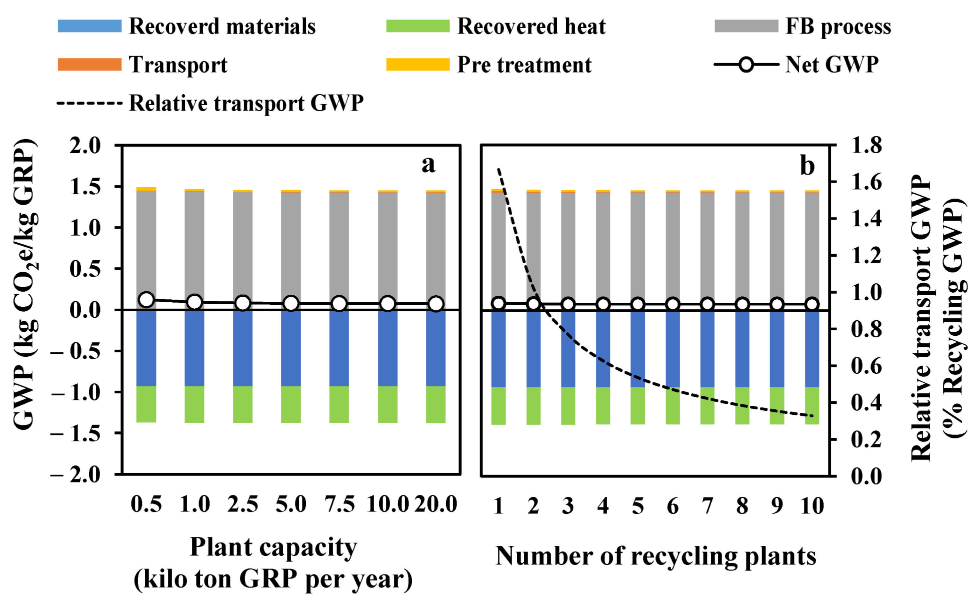

Figure 11. Influence of (**a**) plant capacity alone and (**b**) number of UK plants on FBR GWP for mixed GRP waste.

Figure 11a only assesses the influence of plant capacity (varied from 0.5 to 20 kton per year) and fixes the transportation distance to 200 km. In practice, there is a fixed quantity of GRP waste available; therefore, changing the plant capacity will lead to an increase in the average waste transportation distance (as there will be fewer recycling plants to meet waste demand). To better reflect this relationship, Figure 11b assesses the influence of the number of recycling plants, which determines both the capacity of the plant and the waste transportation distance. In Figure 11b, it is assumed that 81.3 kton of waste GRP is available (as described in Table 1), which is evenly distributed across all plants. The method for assessing transportation distance for a given number of plants is described in [24]. The relationship between number of plants, transportation distance, and plant capacity used as input to Figure 11b is presented in Figure 12.

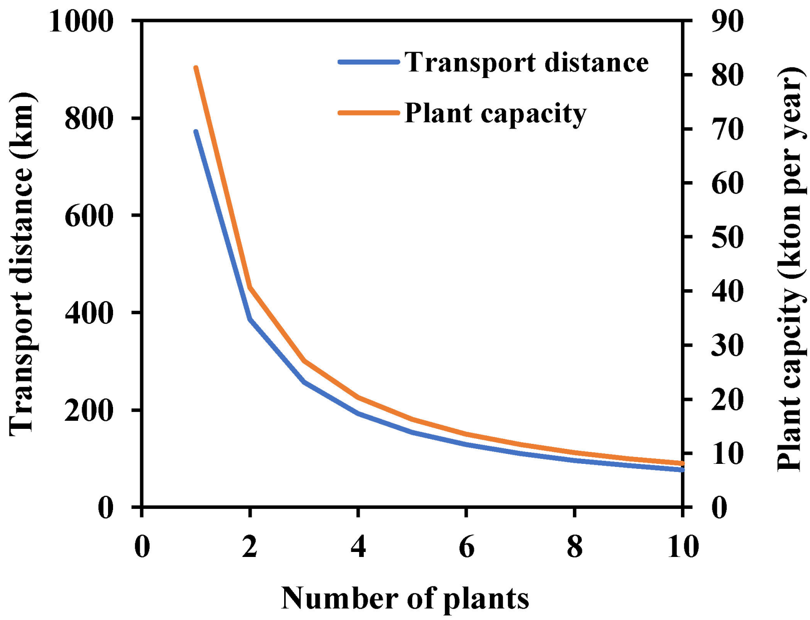

Figure 12. Relationship between number of plants, transportation distance, and plant capacity used as input to Figure 11b.

Given that the main contribution to GWP is GHG emission due to resin oxidation, the output GWP is largely unaffected by the plant capacity in Figure 11a. A slight reduction in GWP is observed with plant capacity due to the increase in natural gas requirements to compensate for higher relative heat loss in smaller plants. Figure 11b also shows the sources and magnitude of GWP associated with UK mixed GRP waste recycling using the fluidised bed as a function of the number of plants. The net GWP is relatively unaffected by the number of plants across the scenarios in Figure 11. The transportation contribution to GWP is shown in Figure 11b relative to the output GWP and is found to account for just 1.7% of the GWP under maximum transportation distance conditions of one recycling plant processing all UK GRP waste.

Continuing to increase the number of recycling plants beyond what is shown in Figure 11 will eventually lead to a rise in GWP, as shown in Figure 13, despite the fact that the reduction in transportation distance enables more localised recycling facilities. This is because (1) a greater number of recycling plants means each plant capacity will be lower to meet demand (see Figure 12), and (2) the GWP associated with the FBR process is inversely proportional to the plant capacity. Therefore, the reduced efficiency of operating small capacity plants ultimately superseding and benefits from lower transport distances and leads to an overall slight growth in net GWP. The minimum net GWP will occur for seven recycling plants, assuming even distribution of GRP waste throughout the UK. The practical challenge of selecting the optimal number of recycling plants to be developed to meet the GRP waste demand requires additional investigation and considerations that fall outside the scope of this research. The plant capacities will most likely be tailored to meet local demand, which will vary spatially across the UK. Figure 13 shows that while this will affect key environmental metrics, the impact is relatively low, and the FBR process can be developed to a range of scales without significantly altering the net GWP of the process overall.

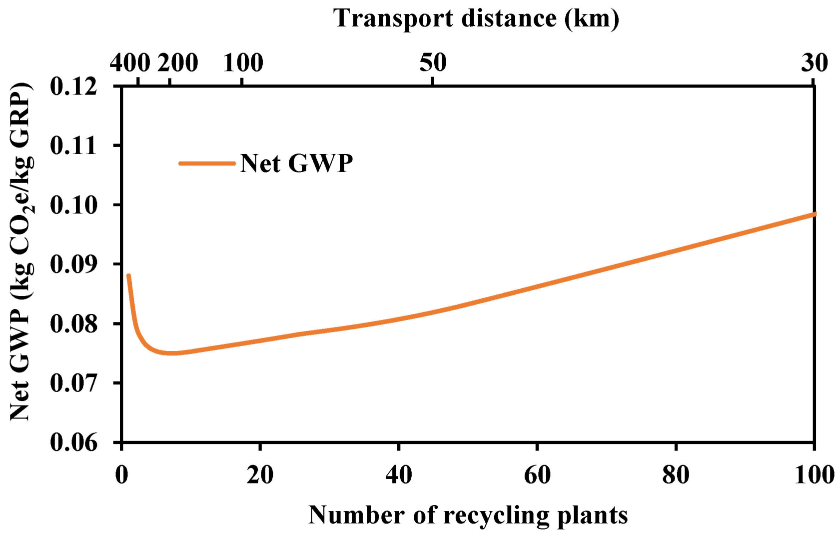

Figure 13. Influence of number of recycling plants on net GWP for mixed UK GRP waste.

3.2. GRP Waste Treatment Comparison

In this section, the GWP associated with the range of EoL GRP solutions are compared. The waste streams considered are as follows: (1) current mixed GRP waste, (2) 2025 EoL–WTB, and (3) 2025 EoL–WTB + Manf. Each of these waste streams has a unique composition and total mass, which are shown in Table 5. It is assumed that only one recycling plant processes the GRP waste streams; therefore, the maximum transportation distance of 772 km is used for the cement kiln and mechanical and FBR processes. Given their ubiquity across mainland UK, transport distance for landfill and EfW scenarios is assumed to be 100 and 200 km, respectively.

Table 5. Composition of the waste streams investigated ([1] assumed to be $CaCO_3$; [2] combination of materials such as paint, adhesive, foam, and balsa).

GRP Waste Stream	Mass (kton per Year)	Weight Fraction in Waste GRP (%)			
		Fibre	Filler [1]	Resin	Other [2]
Mixed GRP waste	81.3	31.9	19.2	48.9	/
EoL–WTB (2025)	10.0	60.0	0	32.0	8.0
EoL–WTB + Manf. (2025)	24.9	43.0	11.7	42.2	3.1

Figure 14a–c show the GWP associated with the various waste treatment processes for current mixed GRP waste, EoL–WTB waste, and EoL–WTB + Manf. waste, respectively. Figure 15 shows a comparison of the net GWP for each waste type and treatment process investigated. The "waste treatment" refers to the net GWP of the process, excluding the GWP offset through material recovery. A negative waste treatment GWP indicates that, regardless of any additional material recovery, the process itself causes a reduction in GWP. This is only observed for cement kiln co-processing and holds for all waste types in Figure 14. The co-processing waste treatment functions by replacing heat energy generation via coke/coal combustion within the cement kiln with an equivalent amount of resin heat energy. The resins used in this study have a slightly higher calorific value than coal; therefore, less mass of resin is required to provide an equivalent amount of heat energy as coal. Furthermore, the GWP associated with resin combustion is lower than that of coal. This means that utilising GRP waste in the cement kiln can offset coal combustion with a lower mass of resin, which itself has a lower GWP associated with combustion. This ultimately reduces the waste treatment GWP by reducing GHG emitted by the cement kiln. The magnitude of the waste treatment GWP is dependent on the resin type and its content within the waste GRP. Increasing resin content within the waste increases the calorific value

of the GRP, meaning more coal can be offset, which lowers waste treatment GWP. Resin types with lower GWP associated with oxidation (such as UPR and vinyl ester) also result in lower GHG emissions for an equivalent heat energy input to the cement kiln.

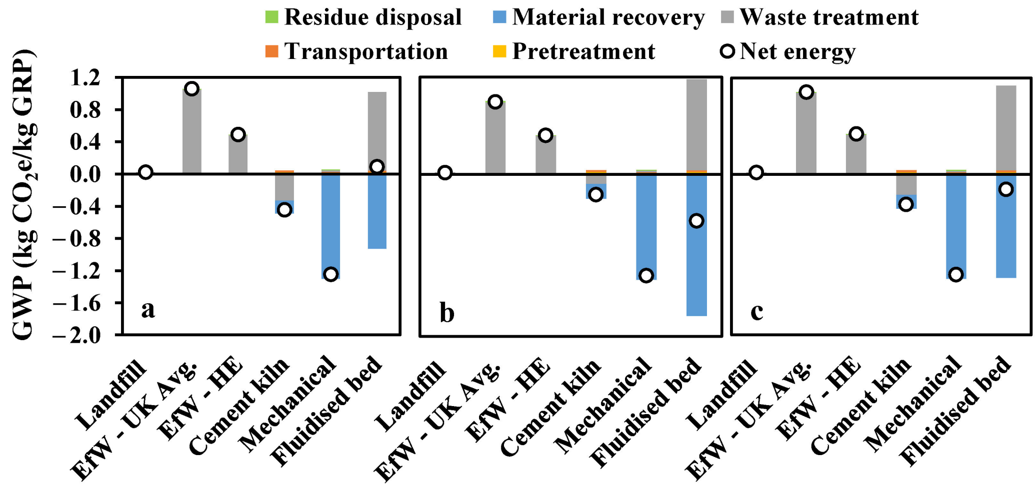

Figure 14. GWP associated with the various waste treatments processes for (**a**) current mixed GRP waste, (**b**) 2025 EoL–WTB, and (**c**) 2025 EoL–WTB + Manf. waste.

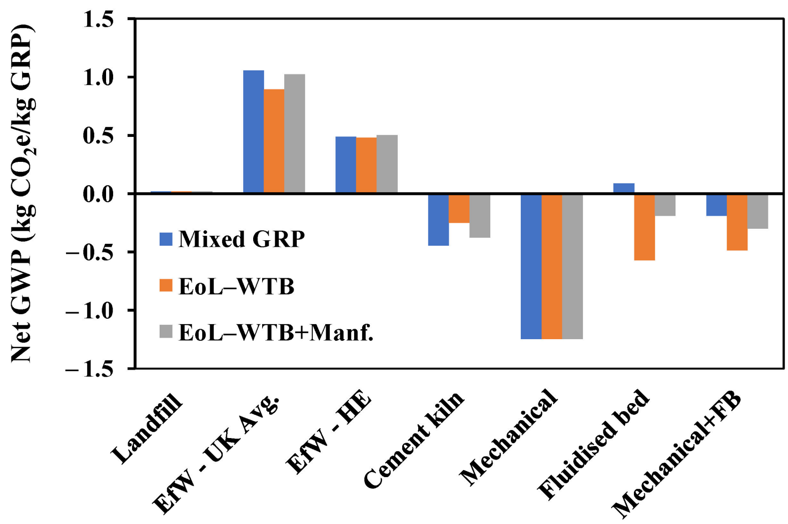

Figure 15. Comparison of net GWP of the various waste treatments processes and waste streams in 2025.

This effect can be seen comparing cement kiln waste treatment GWP for mixed GRP waste and EoL–WTB in Figure 15. The magnitude of waste treatment GWP is less than for mixed GRP waste because EoL–WTB waste has a lower resin content and can, therefore, offset less coal (relative to the mass of GRP processed). Furthermore, epoxy is used in WTB, which has a higher GWP associated with combustion, moving it closer to the GWP of coal oxidation when compared to that of UPR, which is the dominant resin in the mixed GRP waste. Additional material recovery is provided with higher fibre content GRP such as EoL–WTB; however, this does not fully mitigate the higher waste treatment GWP, and overall, the net GWP for the cement kiln process is lower for higher resin content waste, as shown in Figure 15. The chemical composition of the alternative fuel is a factor that influences the cement manufacturer to select a particular alternative fuel for their plant since this impacts the cement production processes itself in addition to the properties of the product [52].

In all cases, processing through EfW results in a positive net GWP regardless of plant efficiency. The UK average EfW plant efficiency has a significantly greater net GWP than all other waste treatments analysed. This is due to the GHG emission associated with resin incineration not being adequately offset, which is a result of low-efficiency energy generation and lack of material recovery facilitated by the process. Lower resin content in the waste GRP results in reduced waste treatment GWP since less GHG emissions are produced relative to the mass of GRP processed. The high-efficiency plant exhibits a lower net GWP by offsetting greater amounts of heat and electricity generation; however, the net GWP remains greater than that of landfilling. In addition to poor GWP outlook, the non-combustible, inorganic fibre and filler fractions of the waste GRP, accounting for 51–60% of the mass of waste GRP in Figure 15, must still be disposed of following processing through EfW.

Figure 15 shows that mechanical recycling consistently has the lowest net GWP of the processes analysed. Unlike other waste treatments, mechanical recycling does not require combustion of the resin fraction in GRP. GWP of waste treatment, therefore, only comes from electricity input to operate the granulation and classification equipment, which is low compared to GHG emissions produced using incineration processes like FBR. In addition to this, GWP offset through recovered material is high since 42% of the mass of the GRP processed is utilised to offset vGF, which itself has relatively high GWP associated with its production [44].

Under most conditions analysed in Figure 15, the FBR process has negative net GWP despite the high levels of waste treatment GWP resulting from resin combustion. EoL–WTB is an ever-growing waste stream in the UK; given the potential environmental benefits shown in Figure 15, the FBR process would be ideally suited to recycle this waste. At present, however, a plant solely processing GRP from EoL–WTB may be able to do so at negative net GWP but would be limited in capacity, which could have economic implications that need to be considered. An EoL–WTB recycling plant may require additional waste volumes from GRP production waste to operate at a sufficient capacity in the near term. Figure 16 shows the net GWP for the various waste treatment routes processing EoL–WTB + Manf. waste streams, which follows the trends observed in Figure 15. Mechanical recycling provides the lowest net GWP across the time scale analysed, with FBR routes being optimal during years with a greater mass of EoL–WTB available.

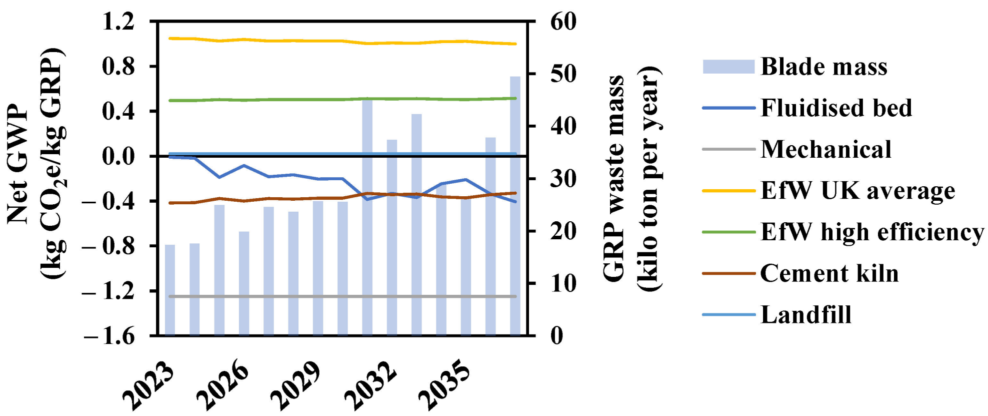

Figure 16. Net GWP of the various waste treatment processes for EoL–WTB + Manf. waste over time.

While both the glass fibres recycled using FBR [42] and the fibrous fractions recycled using the mechanical process [38] have successfully replaced the equivalent mass of vGF in BMC/SMC without impacting mechanical properties, the fraction of vGF they can replace without negatively influencing the resulting mechanical properties differ. Fluidised bed rGF can replace up to 50% of the mass of vGF, whereas fibrous fraction recovered mechanically can only replace 10%. Given that there is finite demand for BMC/SMC products, which have relatively low fibre fractions regardless, this limits the amount of vGF that can be offset. The replacement rate is five times higher for fluidised bed rGF compared to fibrous recyclate obtained mechanically; therefore, there is a greater market potential for reusing this material. This is shown in Figure 17, which shows the fraction of fibrous material that can be reused in the production of new BMC/SMC given the UK demand for these materials (4.8 kton of GF per year) as a function of the percentage of UK mixed GRP waste that is recycled (total UK mixed GRP waste mass is equal to 81.3 kton per year). Figure 17 shows that the UK demand for glass fibre in BMC/SMC products is satisfied by each of the recycling processes at relatively low recycling rates. Given the disparity in vGF replacement rates, however, this occurs at much lower recycling rates for mechanical recycling (1.4%/1.4 kton of GRP per year) compared to fluidised bed (9.7%/7.9 kton of GRP per year), as shown in Figure 17. This shows that while important environmental metrics such as net GWP for mechanical recycling in Figure 15 are superior, this is only because of high material recovery, which might, in fact, be significantly limited by market demand and low replacement rates.

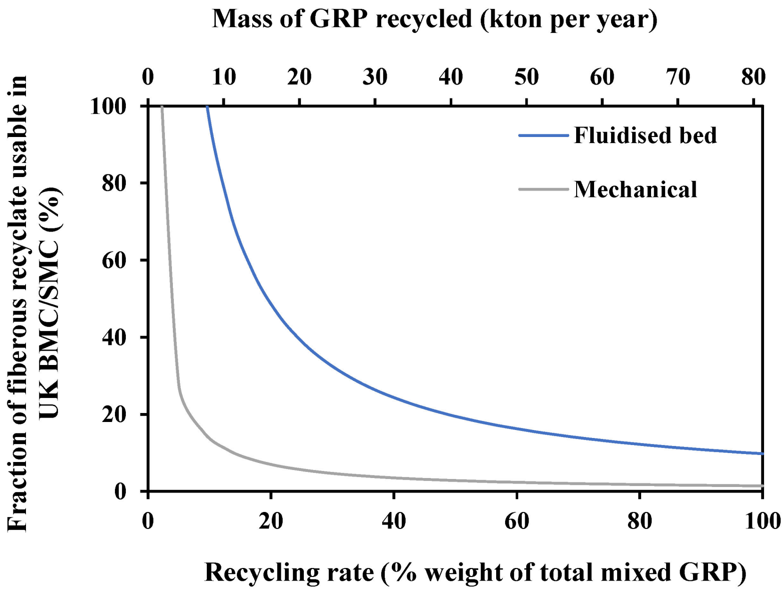

Figure 17. Fraction of fibrous recyclate usable in UK BMC/SMC production for each of the recycling processes.

4. Discussion

In attempting to reduce the carbon footprint and increase the circularity of waste GRP products, traditional disposal routes such as landfilling and EfW should be avoided. While the inert nature of GRP means that the GWP associated with landfilling GRP waste stream is low, this is not a long-term sustainable solution.

Austria, Finland, Germany, and the Netherlands already have legislation that bans the landfilling of GRP wind blade waste, with Wind Europe calling for a Europe-wide ban on landfill blades [53]. This work has shown, however, that while using GRP as feedstock in EfW facilities can reduce landfill burden, this results in significantly higher GWP than simply landfilling. This has also been shown to be the case in North America [13] and China [14]. Given the UK (and other global) commitments to Net Zero 2050, incinerating GRP waste for energy alone must also be avoided. Moreover, legislation banning the landfilling of GRP waste is likely to result in greater production of GHG emissions, where incineration for energy is the only other waste treatment option. Both reducing landfill volumes and GHG emissions are critical efforts that should be pursued. However, legislators should ensure that solutions that can deliver against both targets are mandated and, therefore, evaluate the negative consequences of enforcing landfill bans where low-carbon recycling solutions do not exist at scale.

Cement kiln co-processing is the most established "recycling" route for EoL–WTB waste, with commercial operations in North America [54] and Germany [55]. Current commercial co-processing of GRP waste utilises only EoL–WTB as input material, which provides consistency in the material composition [22]. To the best of the authors' knowledge, mixed GRP waste has not yet been utilised as raw material for clinker production, which will likely present additional uncertainty in waste composition. A recent position paper from major European market players (published by the European Composites Industry Association) called for a strong supporting EU regulatory framework to scale up cement co-processing of waste composites [56]. In doing so, they are calling for European policy makers to recognise co-processing as a recycling process according to the Waste Framework Directive 2008/98/EC. It is important to note that while cement kiln co-processing was found to be carbon-negative for all GRP waste streams in this study (as well as in other publications [22,23]), this is a result of the avoided burden associated with petroleum coke or coal combustion in the kiln. The development of green cement is critical for the decarbonisation of the construction sector, and GRP scrap as an alternative lower carbon fuel source could play a role. However, in kilns where alternative fuels, such as solid recovered fuels, are already being used, it is yet to be established if substitution with GRP scrap will be environmentally preferable. It is therefore critical that further investigations assess the environmental risk of GRP being used to replace other alternative fuels in kilns and whether co-processing of GRP is a sustainable long-term solution.

This study has shown that mechanical recycling could be a low-carbon solution to mixed GRP waste streams, including wind blade waste. This is in close agreement with other studies that investigated the impact of GRP mechanical recycling in North America [13] and China [14]. Unlike FBR, which utilises all the waste GRP mass, the "coarse" fraction remains following the classification stage of mechanical recycling. This fraction accounts for 28% by weight of mechanical recyclate, and to date, no use has been identified and is therefore assumed to be landfilled, which is not a long-term solution. In regions with existing landfill bans, incineration of this waste fraction is anticipated. Given that this coarse fraction is resin-rich, incineration is expected to result in a significant increase in GWP for this waste treatment strategy. As such, it is critical that solutions to utilise all the mechanical recyclate fractions are pursued.

This investigation has shown that an FBR plant could be developed on a commercial scale while maintaining a route to market for all rGF products in the UK. However, FBR only maintains a positive environmental impact by offsetting vGF production. Given that the demand for glass fibre in BMC/SMC production is saturated at relatively low recycling rates, additional applications for rGF must be identified; otherwise, the FBR is not an environmentally sustainable option for all UK GRP waste. The loss in tensile strength [57] and discontinuous, non-aligned, and "fluffy" architecture of fluidised bed rGF [19] will certainly restrict the route to market of these as reinforcement materials, potentially limiting their use to lower grade composite components with lesser market value. Recent advances have been made in rGF regeneration, showing significant improvements in strength and surface functionality can be achieved [15], which would likely diversify the applications available to these materials. These treatments will incur additional resources and environmental impacts that are not included in this model.

This investigation has highlighted that the greatest source of GHG emissions from FBR GRP is CO_2 produced through oxidation of the organic faction in waste feedstocks. As such, it is recommended that future development strategies for FBR decarbonisation focus on utilising inert atmospheres within the reactor, as has been performed for pyrolysis fluidised bed recycling of neat plastics [58]. This has the potential to mitigate direct GHG emissions while collecting secondary organic products, such as pyrolysis oils and syngas [59].

This study utilised data obtained from the operation of The University of Strathclyde's fluidised bed recycling technology, which informed the development, assumptions, and inputs into the energy model. It is expected that data from a larger-scale operation will enhance the precision of modelling and impact assessments. The University of Strathclyde is currently in the process of upscaling the fluidised bed recycling process for wind blade waste. Following the commissioning of the pilot scale plant, upcoming efforts will concentrate on gathering additional data and fine-tuning the models used for environmental impact assessments.

5. Conclusions

This work investigated the GWP of various GRP waste solutions to inform future development and deployment of low GHG emissions GRP recycling technologies. The following conclusions are made:

- Both mechanical and fluidised bed recycling technologies can reduce the GWP compared with conventional disposal routes due to the ability to offset raw material/ energy production.
- FBR was found to be a promising recycling process for EoL–WTB due to relatively low resin content and large material recovery offsets due to the high glass fibre content of this waste stream. It was found that an FBR plant processing EoL–WTB and GRP manufacturing waste could operate at GWP lower than both landfill and EfW and is, therefore, a preferred end-of-life solution over conventional disposal routes.
- Mechanical recycling was estimated to have the lowest GWP of all treatments analysed, resulting from low GHG emissions associated with the process itself and potentially high offsets by replacing glass fibre in the production of BMC/SMC. Limited market size and poor replacement rate, however, may restrict the reuse of mechanical recyclates as reinforcements in secondary composites.
- Without finding additional recyclate routes to market, both mechanical and FBR could not be a solution for all UK composite waste. The environmental assessment of these technologies hinges on the ability to offset raw material production; therefore, it should be reevaluated when considering alternative applications for fibrous recyclates.

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Conflicts of Interest: The authors declare no conflict of interest.

Glossary

BMC	Bulk moulding compound
EfW	Energy from waste
EoL	End of life
EoL–WTB	End-of-life wind turbine blade
EoL–WTB + Manf.	End-of-life wind turbine blade and manufacturing waste
FBR	Fluidised bed recycling
GHG	Greenhouse gas
GRP	Glass fibre-reinforced polymer
GWP	Global warming potential
LCA	Lifecycle assessment
rGF	Recycled glass fibre
SMC	Sheet moulding compound

TRL	Technology readiness level
UPR	Unsaturated polyester resin
vGF	Virgin glass fibre

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